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Overcoming restoration thresholds and increasing revegetation success for a range of canopy species in a degraded urban Mediterranean-type woodland ecosystem

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Widespread decline of Mediterranean-type ecosystem (MTE) woodlands can result in a loss of soil- and canopy-stored seed banks. This can drive woodlands across a biotic threshold, where natural regeneration cannot occur. Without management intervention, these woodlands will suffer local extinction. Using a Mediterranean-type, degraded woodland as a case study, we undertook field trials over 3 years, with the aim of increasing revegetation success by (1) introducing propagules of key canopy species to overcome this biotic threshold and (2) applying commonly used revegetation treatments (abiotic treatments such as the addition of nutrient and water resources, two types of tree guards, and combinations of these). We found that (1) control plants had low establishment success, confirming the crossing of a biotic threshold and the practical irreversibility of the degraded state without intervention, (2) plant establishment was often significantly higher for treated than for control seedlings and (3) supplementation of nutrient and water resources seems to be critical in terms of increasing early seedling establishment for some species. We suggest that in declining woodlands that have crossed biotic thresholds, merely adding propagules does not ensure successful revegetation. The present study has practical implications for restoration activities in degraded MTE communities where biotic thresholds may have already been crossed.

Keywords: invasive species. nutrients. establishment, reforestation; irrigation, tree seedlings; woodland restoration; seedling survival;

Wide-spread decline in the health of Mediterranean-type ecosystem (MTE) woodlands can result in low levels of natural regeneration. We undertook field trials over 3 years, with the aim of increasing revegetation success, finding that the mere addition of propagules did not ensure successful revegetation and that the addition of nutrient resources seems to be critical in terms of increasing early seedling establishment in some species. Without this type of information and management-intervention activities, such woodlands could suffer local extinction.

Introduction

The concept of thresholds in restoration suggests that options for restoration are determined by the current state of the system in relation to biotic and abiotic thresholds (Hobbs and Harris 2001). These thresholds prevent the system from returning to a less-degraded state without some form of management intervention (Hobbs and Norton 1996). State and transition models (Yates and Hobbs 1997; Standish *et al*. 2008) can be used to map a series of thresholds that have to be overcome by active management intervention to allow restoration of woodland structure and function. In addition, there is a need to explore management-intervention techniques that will overcome these thresholds (Yates *et al*. 2000).

There are two types of thresholds that ecosystems can cross, namely, biotic and abiotic. Biotic thresholds are created by factors including lack of propagules, dispersal barriers, herbivory, competition for light, water and nutrients from invasive species, and altered mutualisms (e.g. mycorrhizal fungi and insufficient pollination). Abiotic thresholds result from severely changed abiotic conditions (e.g. soil erosion, altered hydrological conditions, low levels of substrate fertility, low soil-water availability and soil compaction). A crossing of a biotic threshold results in novel combinations of species and functional groups, whereas crossing of an abiotic threshold results in novel abiotic conditions, where species from the original natural ecosystem may find it difficult to establish, even when the biotic threshold, such as a dispersal barrier has been overcome (Vitousek *et al*. 1997; Levine *et al*. 2003; Hobbs *et al*. 2006).

If only biotic thresholds have been crossed, intervention may require only improved management, such as reinstating ecological processes such as fire, to facilitate natural regeneration. For example, in many Australian MTEs, some species such as temperate eucalypts recruit following large-scale disturbances such as fire (Yates *et al*. 1994; Ruthrof *et al*. 2003), and the use of prescribed and well planned fire could facilitate natural regeneration. However, reinstating fire may not be appropriate for many sites. First, if the system is highly degraded, it may have little or no seed store (thus having crossed a biotic threshold) and the post-fire environment may not facilitate natural recruitment. Second, a range of serotinous woody perennials disperse only short distances, and although there may be a few seed-bearing plants in the vicinity, they are unlikely to produce the level of natural recruitment needed to replace senescing adults across the population (Standish *et al*. 2007). Third, the timing and intensity of prescribed fires are often dictated by social, rather than ecological, requirements, so there is a risk that societal pressures will result in low-intensity fires for fuel reduction and may not be adequate to facilitate recruitment of key species or lead to extirpation of species. In these cases, the use of fire to facilitate natural regeneration may fail or facilitate significant shifts in the composition and structure of the plant communities that then require further management intervention.

If biotic thresholds have been crossed and the amount of propagules *in situ* is insufficient to reinstate plant populations, then land managers need to undertake a large-scale input of propagules, such as by planting seedlings, divisions or cuttings, or broadcasting seed. Although the planting of seedlings is generally more successful than broadcast seeding (Standish *et al*. 2008; Ruthrof *et al*. 2010), planted seedlings experience varying levels of establishment success, particularly in MTEs with long drought periods (Kozlowski and Davies 1975; Roche *et al*. 1998). Perhaps seedling mortality could be reduced by understanding thresholds that prevent early seedling establishment and by providing abiotic resources that seedlings would normally receive during natural recruitment (Ruthrof *et al*. 2010).

We suggest that in ecosystems that have already crossed biotic thresholds, forcing those systems back towards a functioning ecosystem through the mere addition of biotic resources (e.g. seedlings) may not be enough to ensure successful restoration. However, the addition of abiotic resources, such as water and nutrient resources, applied to individual plants, may increase establishment success. To this end, we pose the following question, by using a degraded urban woodland in the south-western Australian biodiversity hotspot as an example: is it possible to reintroduce seedlings to degraded urban woodland and increase early establishment by providing abiotic resources to overcome restoration thresholds?

Materials and methods

Study site

The study site of Bold Park is a large (437 ha), highly disturbed, urban *Banksia–Eucalypt* woodland remnant, located 8 km west of the Perth central business district, Western Australia. The region has a warm Mediterranean-type climate with hot dry summers and cool, wet winters. The most complete and nearest weather station and dataset is from the Subiaco Treatment Plant (Station Number 009151), ~2 km from the site, which receives an average annual rainfall of 704 mm, 80% of which falls between May and September (BOM 2012). The region is experiencing a sustained and substantial shift to drier and warmer conditions, specifically, a pronounced, long-term decline in rainfall since the mid-1970s and an increase in average temperatures at a rate of 0.15°C per decade over the same time period (Bates *et al*. 2008).

The major soil type of the park is classified as part of the Spearwood Dune System, consisting of variable depths of siliceous, brown and yellow leached sands derived from an underlying aeolianite of Tamala Limestone (McArthur and Bettenay 1974; Gozzard and Mouritz 1989; McArthur 1991). Nine native-plant communities have been identified in Bold Park, with a total of 568 species (336 native and 232 introduced) (Barrett and Tay 2005), the endemic species contributing to making the south-west of Western Australia one of the 35 global biodiversity hotspots (Mittermeier *et al*. 2011). The trials were undertaken in the *Banksia* woodland community which is characterised by *B. attenuata*

R.Br. and *B. menziesii* R.Br. and scattered emergent *Eucalyptus gomphocephala* DC. trees. The understorey is composed of diverse sclerophyllous shrub layer (Beard 1989).

Bold Park is one of the oldest remnants in the Perth Metropolitan Area and has been used in the past for several land uses. As such, the remnant is affected by weed invasion, feral animal activity (rabbit grazing and burrowing), recreational pursuits including horse riding, and frequent fire events caused by arson. Invasive species dominating Bold Park include species from other MTE regions, such as veld grass (*Ehrharta calycina*), rose pelargonium (*Pelargonium capitatum*), Geraldton carnation weed (*Euphorbia terracina*) and bridle creeper (*Asparagus asparagoides*) (Close *et al*. 2009; Fisher *et al*. 2009). Regeneration of the major canopy species is sparse, perhaps as a result of a combination of degrading factors, including competition from invasive species and a high fire frequency (Fisher *et al*. 2009).

A range of field trials using a randomised block design was established over 3 years, to investigate the effects of plant treatments on the survival and growth of seedlings from a range of canopy species (*Eucalyptus gomphocephala*, *E. decipiens*, *Corymbia calophylla*, *Banksia attenuata* and *B. menziesii*), according to availability of seedlings. All field-trial sites were located randomly within canopy gaps to reduce the effects of adult plants (e.g. competition from large trees for light, nutrients and water) and were located in close proximity to mature trees (live or dead), with no nearby recruitment. Fusilade (Fluazifop-p-butyl, Syngenta Crop Protection Pty Ltd, Pendle Hill, NSW, Australia) was broadcast-applied (4 L ha^{-1}) for *Ehrharta calycina* and Glyphosate (N-phosphonomethylglycine, Monsanto Australia Ltd) (3 L ha^{-1}) was spot-sprayed for *Pelargonium capitatum* and *Asparagus asparagoides* control at least 4 weeks before planting in winter, following the start of the winter rains (June or July). Seedlings were grown from provenance seed for 6 months before planting at a local nursery. The trials were established by experienced planters using planting tools (Pottiputkis, Lannen Plant Systems, Säkylä, Finland), at a density of approximately one plant m^{-2}, being approximately half the density of seedlings observed after natural recruitment events (Ruthrof 2001; Ruthrof *et al*. 2003) to reduce the risk of competition. Given that the first year is the most critical period for the establishment of planted seedlings (Savill *et al*. 1997; Benayas *et al*. 2002; Castro *et al*. 2004), monitoring for survival and growth was undertaken 1 year after planting.

Year-1 trials

A field trial was established in July 2002 to test the response of *E. gomphocephala* seedlings to five treatments, using blocks measuring 30 m × 6 m, each of which contained five plots of 6 m × 6 m. The blocks were replicated three times at each of three sites within Bold Park (Table 1). The treatments allocated randomly to the plots in each block were as follows: control, fertiliser, soil wetter, plastic tree guards, and soil wetter + fertiliser + plastic tree guards (Table 2). Thirty five *E. gomphocephala* seedlings were planted into each plot (a total of 1575 seedlings).

The same trial was established for *B. menziesii* at one site (Table 1). This trial used blocks measuring 16 m × 4 m, each of which contained four plots of 4 m × 4 m. The blocks were replicated three times at the site. The following four treatments were chosen to suit this proteaceous, P-sensitive species: control, plastic tree guards, low P fertiliser, plastic tree guards + low P fertiliser (Table 2). Twenty seedlings were planted into each plot (a total of 240 seedlings).

Year-2 trials

In June 2003, field trials were established to test the response of *E. decipiens*, *E. gomphocephala*, *B. attenuata* and *C. calophylla* seedlings to treatments, using blocks measuring 20 m × 5 m, each of which contained five plots of 4 m × 5 m. The blocks were replicated three times at each site (Table 1). The treatments allocated randomly to the plots in each block were as follows: control, plastic tree guards, season extension watering, season extension watering + plastic tree guards, and monthly watering throughout summer (Table 2). The 'season extension' watering treatment was intended to mimic a longer winter season; thus, watering was applied from November until December. The monthly watering throughout summer began at the beginning of November as temperatures increased and continued until the following winter rains commenced in May. Twenty seedlings were planted in each plot (totalling 600, 600, 900, 600 plants for *E. decipiens*, *E. gomphocephala*, *B. attenuata* and *C. calophylla*, respectively).

Year-3 trials

In June 2004, trials were established to test the response of *E. gomphocephala*, *B. attenuata* and *B. menziesii* seedlings to treatments, using blocks measuring 20 m × 5 m, each of which contained four plots of 4 m × 5 m. The blocks were replicated three times at each site (Table 1). The following four treatments were allocated randomly to the plots in each block: control, shade cloth tree guards, fertiliser, and shade cloth tree guards + fertiliser (Table 2). Shade-cloth bags were tested previously, and reported to have benefit (Close *et al*. 2009). In total, 22, 30 and 30 seedlings were planted in each plot for *E. gomphocephala*, *B. attenuata* and *B. menziesii*, respectively, totalling 528, 360 and 360 plants, respectively.

Statistical analysis

All data were analysed using R software environment (R 2011). Survival data were analysed using a binomial generalised linear model, checking for over-dispersion, whereas height data were analysed using a linear model for surviving plants only, with checks for normality of residuals and equality of variances among groups. Year-1 data on *B. menziesii* were analysed considering separate effects for guards, bags and fertiliser, whereas Year-1 data on *E. gomphocephala* were analysed considering separate effects for each treatment and an additional effect for all three treatments in combination. Year-2 data were analysed considering separate effects for guards, season extension and watering through summer, for all species. Year-3 data were analysed considering effects for guards and fertiliser. All interactions among treatments were considered where appropriate. In all cases, a full model was fitted first and then model simplification was conducted, with stepwise removal of non-significant terms at $P = 0.05$, until only terms significant at $P < 0.05$ remained. P-values were obtained using chi-square tests for binomial data and F-tests for height data. Where required, P-values for pair-wise comparisons were obtained through pooling the relevant groups and testing for significant reduction in explanatory power using the chi-square test or F-test as appropriate.

Results

Year-1 trials

For *B. menziesii*, a very high overall mortality meant that there was no significant difference in survival or growth rates as a result of treatment. Despite high overall mortality, survival of *Eucalyptus gomphocephala* seedlings differed with treatment ($P = 0.005$), with seedlings receiving all three treatments (fertiliser, soil wetter and tree guards) having higher survival (8.6%) than seedlings with just one treatment (3.5%, $P < 0.001$) (Table 3). There was also a significant ($P < 0.001$) site effect. The recently burnt site, Mount Claremont, showed high levels of survival, compared with the other two sites. These seedlings treated with fertiliser and with the combination (soil wetter, fertiliser and tree guards) also had significantly higher growth rates after 1 year than did the control ($P = 0.01$ and $P = 0.005$, respectively) (Table 4).

Year-2 trials

For *B. attenuata*, survival was improved by watering through summer ($P < 0.001$), but not by guards or season extension (Table 3). There was also a significant ($P = 0.05$) site effect, with higher survival at Mount Claremont. Growth rates were not affected by treatment, but growth rates were greater at Mount Claremont ($P < 0.001$) (Table 4).

For *C. calophylla*, survival was improved by watering through summer ($P < 0.001$), by guards ($P < 0.001$) and by season extension + guards ($P = 0.02$) (Table 3). Growth rates were also improved by guards ($P < 0.001$), but not by season extension or by watering through summer (Table 4). There were significant ($P < 0.001$) site effects for both survival and growth rates, which were lower at Eastern Gateway upslope (Tables 3, 4).

For *E. decipiens*, survival was improved by watering through summer ($P < 0.001$), by guards ($P < 0.001$) and by season extension ($P = 0.02$) (Table 3). Growth rates were also improved by watering through summer ($P < 0.001$) and by guards ($P < 0.001$), but not by season extension (Table 4). There were significant site effects for both survival and growth rates ($P < 0.001$), which were higher at Skyline (Tables 3, 4).

For *E. gomphocephala*, survival was improved by guards ($P = 0.001$), but not by watering through summer or by season extension (Table 3). Growth rates were improved by season extension ($P = 0.009$) and by guards ($P < 0.001$), but not by watering through summer (Table 4). There were significant ($P < 0.001$) site effects for both survival and growth rates, which were higher at veld grass area (Tables 3, 4).

Year-3 trials

For *B. attenuata*, fertiliser by itself greatly reduced survival from 57% to 16% ($P < 0.001$) (Fig. 1). Guards had no significant effect on unfertilised plants but increased the survival of fertilised plants from 16% to 38% ($P = 0.002$). Overall, guards also resulted in an increased height in surviving trees (37 cm *v.* 23 cm, $P < 0.001$) (Fig. 2).

For *B. menziesii*, fertiliser decreased survival from 54% to 43% overall ($P = 0.03$), whereas guards had no significant effect (Fig. 1). Guards resulted in an overall increased height in surviving trees (34 cm *v.* 26 cm, $P < 0.001$) (Fig. 2).

For *E. gomphocephala*, fertiliser increased survival overall from 76% to 84% ($P = 0.02$) and guards increased survival overall from 73% to 87% ($P < 0.001$) (Fig. 1). Guards also increased height of survivors overall from 66% to 70% ($P = 0.007$) and fertiliser increased height of survivors overall from 66% to 69% ($P = 0.05$) (Fig. 2).

Discussion

Degraded ecosystems that have crossed biotic thresholds, that is, have a limited or no capacity to regenerate autonomously, require some form of management intervention such as supplementation of propagules. The present study has demonstrated that revegetation of key canopy species in a degraded, biodiverse woodland is possible and can be successful in terms of increasing seedling survival. However, the addition of propagules such as seedlings alone is often not adequate, given the low survival and growth rates of the control plants. Continued improvements in revegetation techniques are essential, particularly given that revegetation may become increasingly difficult in regions that continue to become hotter and drier (Bates *et al*. 2008). Indeed, the practice and outcomes of ecological restoration will most certainly be altered by the changed biophysical settings that will be prevalent in future climate scenarios (Harris *et al*. 2006). Thus, understanding and overcoming restoration thresholds will become important in regions with decreasing precipitation, particularly the biodiverse Mediterranean regions of the world where climate impacts on biodiversity are predicted to be high (Klausmeyer and Shaw 2009; Yates *et al*. 2010).

Given the relatively low survival rates of seedlings in the control plots, it seems that the addition of propagules (to push the system back across the biotic threshold), without any other form of supplementation, is not be sufficient. There are many abiotic and biotic factors to consider in the natural regeneration of these propagules (Ruthrof *et al*. 2010). For many MTE species, successful recruitment depends on a combination of diverse events and conditions, including the existence of an adequate soil- or canopy-stored seedbank, a disturbance event, some type of dormancy-breaking event, and a viable seed-bed (Bond and van Wilgen 1996; Carrington and Keeley 1999). Particular abiotic and biotic characteristics that facilitate seedling establishment that should be considered in restoration include a pulse of plant-available nutrients (Cummings *et al*. 2007), increased moisture (Loneragan and Loneragan 1964) and a (temporary) reduction in plant competition (Wellington and Noble 1985).

Studies of the effects of nutrients on seedling establishment of canopy species in MTEs have often focussed on eucalypts in old-fields (Schönau and Herbert 1989; Graham *et al*. 2009), mine-sites (Ruthrof 1997; Koch and Samsa 2007), forest (Stoneman *et al*. 1995) and degraded woodlands (Ruthrof *et al*. 2010). Although these studies are useful, the effects of nutrients on the growth and survival of planted seedlings in degraded woodlands from other genera (e.g. *Banksia*) need to be better understood. In our study, the survival of *B. attenuata* and *B. menziesii*, both species being P sensitive (Handreck 1991), was significantly reduced with the addition of nutrients, even though a slow-release fertiliser specifically designed for P-sensitive plants was used. This could be due to several factors, including elevated existing soil P concentrations, or selective grazing of these fertilised seedlings by rabbits. Elevated soil P has previously been recorded in this woodland and stems from long-term changes in vegetation composition, particularly from invasion by *Ehrharta calycina* and *Pelargonium capitatum* (Fisher *et al*. 2006). If rabbits or invasive species were the cause of higher mortality rates, successful revegetation of these key proteaceous elements will be challenging. By contrast, seedlings from other species groups known to respond positively to nutrients, such as *E. gomphocephala*, showed an increase in survival and growth with the addition of nutrients. Clearly, such differential species responses to the addition of nutrient resources need to be understood and treatments refined (e.g. NPK optimisation, concentration and pulse-release characteristics of the nutrient delivery system) before implementing a planting program.

Supplementary water via irrigation to increase seedling establishment has been investigated in restoration, such as in old-field revegetation (Benayas 1998; Graham *et al*. 2009) and reforestation (Badano *et al*. 2009; Estrela *et al*. 2009). For example, in field trials established in Spain, seedling survival was lowest (53%) in the control plots and much higher (93%) for seedlings under irrigation, following 3 years of growth (Benayas 1998). Here, we have demonstrated that for *B. attenuata*, *C. calophylla* and *E. decipiens*, seedling survival increased if watering occurred throughout the summer period. For *C. calophylla* and *E. decipiens*, the addition of tree guards in combination with 'season extension' watering also increased survival. Furthermore, seedling growth in *E. decipiens* and *E. gomphocephala* increased significantly with watering throughout summer and season extension watering, respectively. However, more automated and efficient systems of applying water (other than the costly and invasive hand-watering undertaken in the present study, which also caused some damage to other vegetation by moving hoses) need to be tested, such as trickle irrigation.

Seedlings with tree guards were taller (for *C. calophylla*) and had higher survival rates (for *E. gomphocephala*) than those without, demonstrating the benefits of tree guards for certain species. Those seedlings without guards, particularly *B. attenuata* and *B. menziesii*, were heavily grazed by rabbits, compounding the low natural recruitment and establishment of *Banksia* in this particular urban woodland (Ruthrof 2005). This highlights the need for the development of appropriate control policies for vertebrate herbivores for woodlands undergoing revegetation activities. An additional benefit from tree guards was the significantly taller seedlings in both *Banksia* species in the present study. Although survival is clearly more important than growth, taller seedlings may be more competitive in areas where competition with invasive species is a significant factor negatively affecting establishment. Other studies have shown that tree guards contribute to revegetation efforts by protecting seedlings from vertebrate herbivores (Opperman and Merenlender 2000), improving the microclimate surrounding the seedling (Lai and Wong 2005) and improving the growth of seedlings (Close *et al.* 2005; Ladd *et al.* 2010). For example, a combination of tree guards and weed mats significantly improved the establishment and growth of *Cyclobalanopsis edithiae*, planted in Hong Kong (Lai and Wong 2005). However, concerns have been raised about such tree guards in certain environments, given that they may in fact create an unfavourable microclimate with high temperatures (Close et al. 2009).

Selecting sites for restoration, particularly in urban woodlands, often poses socio-political challenges and often results in a compromise among goals and values for the site, the amount of community support, societal preferences, and the amount of funding, time and labour available to the project (Ruthrof and Valentine 2010). Results from the present study showed that seedling responses were often determined by site. For example, survival and growth of *B. attenuata* were higher in the recently burnt area than in the two unburnt areas for seedlings planted in the same year. Higher survival and growth rates could be a result of reduction in competition or a release of nutrients in the post-fire environment. In addition, survival and growth of *E. gomphocephala* were higher in areas that once had been dominated by the perennial invasive grass *Ehrharta calycina* but where it had been controlled before planting. Perhaps the removal of the dominant invasive species facilitated higher seedling-establishment rates by reducing competition.

There may be an opportunity to utilise created or natural stochastic events (e.g. arson, wildfires or prescribed burns) in revegetation, taking advantage of reduced competition and increased nutrient availability to assist with establishment and growth in planted seedlings. For small urban woodlands with limited resources, it may be beneficial to focus restoration efforts on those key areas. Following these events, herbivore control and revegetation at high densities (e.g. 1000–4000 plants ha^{-1}) could be undertaken, using local species suited to current environmental conditions, and that can compete against invasive species (Fisher *et al*. 2009).

In conclusion, restoration of key structural components in degraded biodiverse woodlands should consider biotic and abiotic thresholds and mimic characteristics of the natural recruitment environment. Our results indicated that without some form of plant and/or soil amendment, seedling establishment success, particularly in long-unburnt areas, will be minimal. The supplementation of abiotic resources (e.g. nutrients) is important for particular species, given the clear species differential response seen in the present study, and protection from vertebrate herbivores is critical for increasing early seedling establishment. Differential site responses within the woodland suggest that appropriate site selection, taking account of natural or induced stochastic events (e.g. arson or wildfires), could assist in increasing revegetation success.

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Table 1. Description of location of field trials, species and treatments

Year	Species	Location	Coordinates	Time since last fire (years)	Treatments
1	*Eucalyptus gomphocephala*	Mount Claremont	31°57′30.66″S 115°46′08.47″E	1	Control, fertiliser, soil wetter, plastic tree guards, and soil wetter + fertiliser + plastic tree guards
		Oceanic Precinct	31°56′28.90″S 115°46′12.16″E	Long unburnt	As above
		Veld grass	31°56′37.37″S 115°46′21.98″E	Long unburnt	As above
	Banksia menziesii	Eastern Gateway	31°56′30.53″S 115°46′42.01″E	Long unburnt	Control, plastic tree guards, low P fertiliser, plastic tree guards + low P fertiliser.
2	*E. decipiens*	Skyline	31°56′15.55″S 115°46′32.41″E	Long unburnt	Control, plastic tree guards, season extension watering, season extension watering + plastic tree guards, monthly watering throughout summer
		South of Pines	31°57′16.05″S 115°46′23.20″E	Long unburnt	As above
	E. gomphocephala	Oceanic Precinct	31°56′28.90″S 115°46′12.16″E	Long unburnt	As above
		Veld grass	31°56′37.37″S 115°46′21.98″E	Long unburnt	As above
	B. attenuata	Mount Claremont	31°57′30.66″S 115°46′08.47″E	2	As above
		Oceanic Precinct	31°56′28.90″S 115°46′12.16″E	Long unburnt	As above
	Corymbia calophylla	Eastern Gateway: upslope	31°56′30.74″S 115°46′42.02″E	Long unburnt	As above
		Eastern	31°56′31.50″S	Long	As above

		Gateway: downslope	115°46′44.81″E	unburnt	
3	*E. gomphocephala*	Hovea Zamia	31°56′41.17″S 115°46′12.18″E	Long unburnt	Control, shade cloth tree guards, fertiliser, shade cloth tree guards + fertiliser
		Eastern Zamia	31°56′40.37″S 115°46′27.80″E	Long unburnt	As above
	B. attenuata	Eastern Zamia	31°56′40.37″S 115°46′27.80″E	Long unburnt	As above
	B. menziesii	Eastern Zamia	31°56′40.37″S 115°46′27.80″E	Long unburnt	As above

Table 2. Description of treatments applied in the revegetation trials

Treatment	Application notes
Control	Seedlings planted without any treatment.
Fertiliser	Nutra-tab tree pellets: % w/w: total N 12.00, total P 3.40, K as sulfate 10.00, Ca as phosphate 3.75, S as sulfate and phosphate 11.80, Mg as oxide 0.60, Cu as sulfate 0.05, Zn as oxide 0.05, Fe as sulfate 0.30, Mn as sulfate 0.20, B as tetraborate 0.01. One fertiliser tablet was placed beneath each root ball.
Soil wetter	Granulated attapulgite propylene oxide–ethylene oxide block polymer. One teaspoon (5 g) was added beneath each root ball at the time of planting.
Plastic tree guards	460 mm high, width of 260 × 260 × 260 × 260 mm (staked into a triangle), placed surrounding each seedling.
Low P fertiliser	Slow-release fertiliser (3–6 month) for native plants. Constituents: N 17.9%, P 0.8%, K 7.3%, S 9.9%, Mg 0.24%. Trace elements (mg kg^{-1}): B 42, Cu 106, Fe 27 882, Mn 124, Mo 40, Zn 35, Cd 0.7, Pb 4. One teaspoon (5 g) was added beneath each root ball.
Shadecloth tree guards	As above but constructed of green wavelength – neutral shade close that intercepts ~54% of incident photosynthetically active radiation (PAR) (see (Close *et al.* 2009)). One guard was placed surrounding each seedling.
Season extension	Watering of ~2 L per seedling once a month, commencing at the beginning of the dry period (November) and watering until December.
Monthly watering throughout summer	Watering of ~2 L per seedling once a month, commencing at the beginning of the dry period (November) and watering throughout the summer period until winter rains began (May).

Table 3. Percentage survival (95% binomial confidence intervals) after 1 year following planting in revegetation trials according to year, species, site and treatment of a range of key canopy species in Bold Park, Western Australia

Year and species	Site	Treatment: Control	Guards	Low P fertiliser	Fertiliser + guards	Fertiliser	Soil wetter	Soil wetter+ fertiliser + guards	Season extension + guards	Season extension
Year 1 – 2002										
Banksia menziesii	Eastern Gateway	8.3 (3–18)	8.3 (3–18)	13.3 (6–25)	5.0 (1–14)					
Eucalyptus gomphocephala	Oceanic Precinct	1.9 (0–7)	0 (0–3)			0 (0–3)	0.9 (0–5)	6.7 (3–13)		
	Veld grass	0 (0–3)	0 (0–3)			0 (0–3)	0.9 (0–5)	0 (0–3)		
	Mount Claremont	19.0 (12–28)	9.5 (5–17)			9.5 (5–17)	15 (5–18)	19 (12–28)		
Year 2 – 2003										
B. attenuata	Oceanic Precinct	5 (1–14)	5 (1–14)						13.3 (6–25)	11.7 (5–23)
	Mount Claremont	13.3 (6–25)	5 (1–14)						18.3 (9–30)	18.3 (9–30)
Corymbia calophylla	Eastern Gateway downslope	33.3 (22–47)	53.3 (4–7)						73.3 (60–84)	58.3 (45–71)
	Eastern Gateway upslope	15.0 (7–27)	11.7 (5–23)						31.7 (20–45)	18.3 (9–30)
E. decipiens	South of Pines	10 (4–20)	15 (7–27)						41.7 (29–55)	25.0 (15–38)
	Skyline	26.7 (16–39)	41.7 (29–55)						43.3 (31–57)	50 (37–63)
E. gomphocephala	Oceanic Precinct	35.0 (23–48)	48.3 (35–62)						40 (28–53)	36.7 (25–50)
	Veld grass	73.3 (60–83)	88.3 (77–95)						85.0 (73–93)	68.3 (55–79)

Table 4. Mean (s.e.) height (cm) after 1 year following planting in revegetation trials according to year, species, site and treatment of a range of key canopy species in Bold Park, Western Australia

Species	Site	Treatment								
		Control	Guards	Low P fertiliser	Fertiliser + guards	Fertiliser	Soil wetter	Soil wetter + fertiliser + guards	Season extension + guards	Season extension
Year 1 – 2002										
Banksia menziesii	Eastern Gateway	25.8 (3.2)	28.2 (1.7)	27.2 (1.7)	28.7 (3.2)					
Eucalyptus gomphocephala	Oceanic Precinct	17.5 (1.5)	n.a.			n.a.	16.0 (n.a.)	38.7 (3.5)		
	Veld grass	n.a.	n.a.			n.a.	33.0 (n.a.)	n.a.		
	Mount Claremont	28.1 (1.8)	32.8 (3.3)			40.2 (2.3)	34.8 (2.3)	37.4 (3.2)		
Year 2 – 2003										
B. attenuata	Oceanic Precinct	17.0 (2.1)	18.3 (3.7)						16.0 (1.5)	15.0 (1.6)
	Mount Claremont	20.5 (2.9)	19.7 (0.7)						25.7 (3.2)	19.3 (1.8)
Corymbia calophylla	Eastern Gateway downslope	38.6 (3.5)	41.4 (2.2)						45.4 (2.8)	36.7 (1.7)
	Eastern Gateway upslope	22.2 (1.3)	28.6 (2.9)						35.2 (1.7)	25.3 (1.3)
E. decipiens	South of Pines	32.8 (3.8)	50.2 (2.8)						44.6 (2.2)	38.1 (3.6)
	Skyline	44.9 (3.9)	46.1 (3.6)						46.0 (3.4)	39.3 (1.8)
E. gomphocephala	Oceanic Precinct	44.6 (2.3)	53.9 (1.4)						53.2 (1.7)	45.4 (1.9)
	Veld grass	68.4 (2.4)	74.4 (1.8)						84.6 (2.1)	71.0 (2.1)

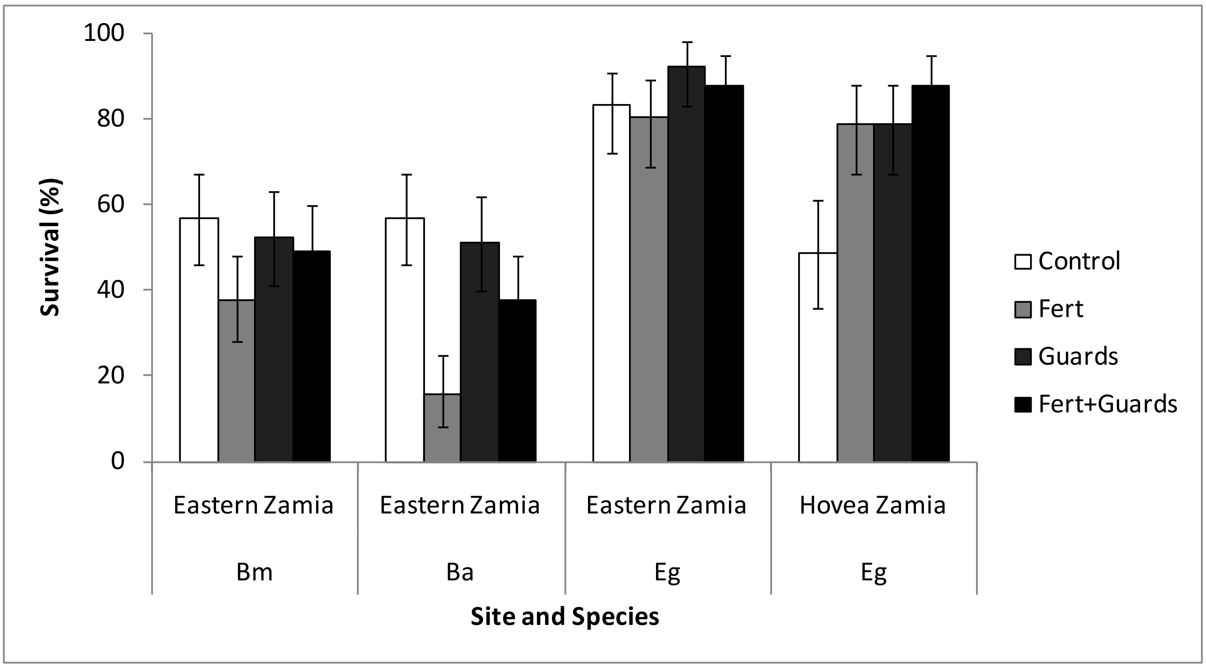

Fig. 1. Total survival (%) (with 95% confidence intervals) for *Banksia menziesii* (Bm), *B. attenuata* (Ba) and *Eucalyptus gomphocephala* (Eg) seedlings after 1 year of growth with four revegetation treatments (control, fertiliser, tree guards + fertiliser + tree guards) at two sites (eastern Zamia and Hovea Zamia) in Bold Park, Western Australia.

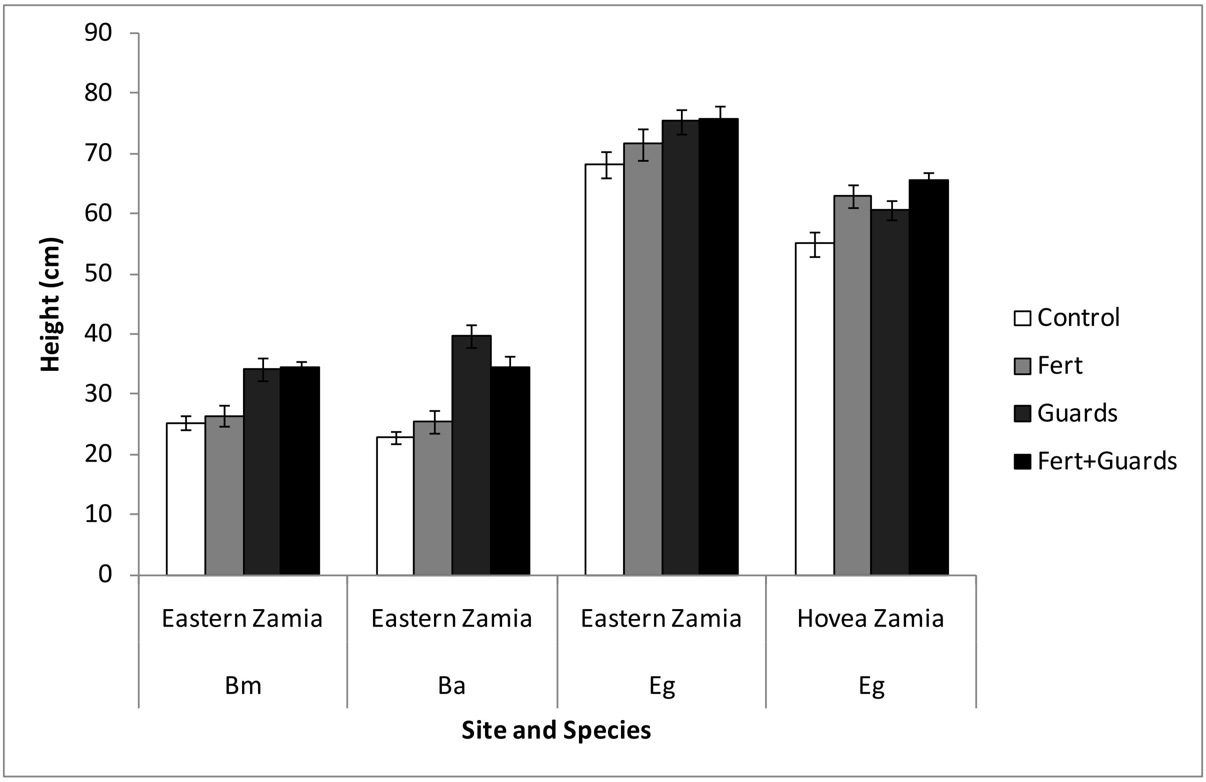

Fig. 2. Mean (±s.e.) height (cm) of *Banksia menziesii* (Bm), *B. attenuata* (Ba) and *Eucalyptus gomphocephala* (Eg) seedlings after 1 year of growth with four revegetation treatments (control, fertiliser, tree guards + fertiliser + tree guards) at two sites (eastern Zamia and Hovea Zamia) in Bold Park, Western Australia.